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The Impact of Prolonged Flooding on the Vegetation of Coomalbidgup Swamp, Western Australia.

R H Froend[1] & P G van der Moezel[2]

[1]Surface Water Branch, Water Authority of Western Australia, PO Box 100 Leederville, WA 6007
[2]Alan Tingay and Associates, 35 Labouchere Rd, South Perth, WA 6151.

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Abstract

A preliminary study of the impacts of prolonged flooding on the vegetation of Coomalbidgup Swamp, an ephemeral wetland near Esperance, Western Australia, identified changes in composition and physiognomy of dryland and wetland vegetation. A high mortality was observed after flooding for all species typically part of the surrounding dryland flora. Some stands of wetland and dryland vegetation have been partially inundated for up to 6 years, which has resulted in 100% mortality of dryland species. The wetland species, *Melaleuca cuticularis* and *Eucalyptus occidentalis*, showed tolerance towards these conditions, although 45.5% of the individuals that were healthy prior to flooding were either dead or dying after prolonged flooding. There was no apparent relationship between tree vigour and age.

Melaleuca cuticularis and *Eucalyptus occidentalis* seedlings emerged around the wetland margins within months of the water levels receding. Little recruitment of native dryland species was evident even 2 years after water had receded. However, extensive weed invasion and establishment of *E. occidentalis* and *M. cuticularis* in this zone contributed to a change in composition of the peripheral vegetation. Secondary salinity and the extent of flooding within the catchment is likely to increase, leading to further degradation and change in wetland vegetation. The study provides an insight into the possible effects of altering the flooding regime of ephemeral wetlands.

Introduction

The species composition of wetland vegetation is influenced primarily by water regime, the key parameters being water depth, flooding frequency and duration. Flooding tolerance varies depending on species, age and quality of floodwaters, however rhizosphere oxygen deprivation is eventually fatal to all species irrespective of their flooding tolerance. Even among wetland species, there are no known cases of any prolonged survival over weeks or months of roots being entirely deprived of oxygen (Crawford 1992). Death of wetland plants, as a consequence of prolonged flooding, accelerates wetland degradation because of a reduced or ineffective buffer to nutrient input, reduced evapotranspiration resulting in increased capillary rise of saline water, destabilisation of sediment, and loss of food source and habitat for fauna.

Changes in water regime and water quality that lead to the decline of wetland vegetation are usually associated with disturbances within the catchment but are external to the wetland (Froend *et al.* 1987; Froend & McComb 1991). Removal of native dryland vegetation results in increased groundwater recharge, associated increased mobility of salt stored in sub-soils, and increased salt and nutrient concentration in the wetland. There have been few south-western Australian studies which have examined the process of wetland vegetation degradation due to salinity and waterlogging, and/or nutrient enrichment, although degradation is widespread in the south-west (Halse 1993). Of the Western Australian rural wetlands studied to-date, *e.g.* Lake Toolibin (Mattiske 1978; Froend *et al.* 1987; Halse 1987; Anon. 1987; Bell & Froend 1990) and Lake Towerrinning (Froend & McComb 1991), the development of secondary salinity and waterlogging within the catchment has been well advanced by the time of study. Little is known about the changes that wetland plant communities undergo during the early part of the process of secondary salinisation and waterlogging. Without baseline data collected before or during the early phases of vegetation change, it is difficult to understand the ecological processes involved. At present we rely largely on historical information to determine the sequence of events which lead to environmental change and degradation.

Of particular importance is the impact that increased salinity and waterlogging has on the recruitment of wetland plants. Most species are dependent on seed production, germination and seedling establishment for successful recruitment, although sedges, rushes and submerged macrophytes readily reproduce by vegetative means (Froend *et al.* 1993). If increased salinity and waterlogging results in the death or degradation of mature individuals at lower elevations, then the continued survival of the population is dependent on the successful recruitment of seedlings at higher elevations. The effect of salinity and waterlogging on germination at higher elevations in the species range at a wetland is therefore critical.

On the south coast of Western Australia near Esperance, a combination of extensive clearing and above-average rainfall years (1986, 1989) caused increased groundwater recharge and surface water retention to an extent where natural ephemeral water courses and basins have retained surface water for prolonged periods. One ephemeral wetland, Coomalbidgup Swamp, has reportedly contained surface water since the winter of 1986. During 1989, winter rainfall in the catchment was heavy, causing severe water erosion and flooding in the catchment (Anon. 1990), and increased water levels in the swamp. Mortality of fringing wetland and dryland vegetation has resulted due to prolonged inundation.

Coomalbidgup Swamp provides an opportunity to examine changes in wetland vegetation structure and composition as a consequence of unusually high water levels and a prolonged flooding period. As the catchment had been cleared in relatively recent times (since 1964), the study permitted the examination of wetland plant community response at a relatively early phase of altered catchment history (*cf.* wheatbelt catchments). The aspects of wetland vegetation dynamics that were investigated include the extent of vegetation mortality as a result of the 1986 and 1989 flood events, the success of seedling recruitment, and the colonisation of the swamp banks as water levels receded.

Study Area

Coomalbidgup Swamp is situated approximately 45 km west of Esperance, Western Australia. A single intermittently-flowing creek drains an area of 97 km^2 and empties into the swamp from the north-east (Fig 1). Most of the Coomalbidgup catchment (approx. 95%) was cleared for agriculture during 1964-1972. Small areas of remnant vegetation exist primarily along water courses and around wetland basins, and the vegetation surrounding Coomalbidgup swamp represents the largest stand of native vegetation in the catchment. Secondary salinity is not currently a major concern within the catchment, however, research by the Western Australian Department of Agriculture (Bob Nulsen, *pers. comm.*) indicates that subsoil salt stores in the catchment are extensive, and pronounced salinisation will occur in the near future with continued elevation of groundwater levels.

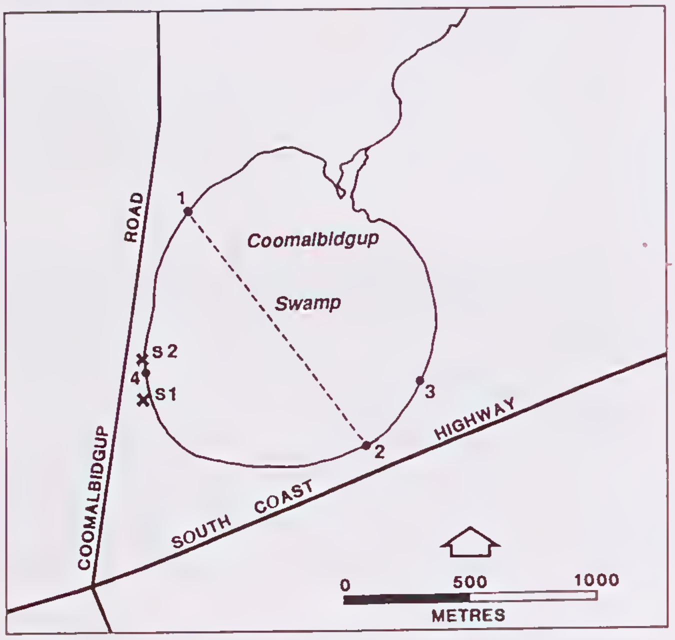


Figure 1. Location of sample sites at Coomalbidgup Swamp. Sites 1, 2, 3 and 4 represent the shore transects, and S1 and S2 the seedling plots. The traverse transect is shown by the dotted line between shore transect 1 and 2.

The area has a 25 year average (1965-89) annual rainfall of 480 mm. The annual rainfall during 1989 was 35% above the average, and contained the wettest two, three, and four consecutive months recorded since 1969. Runoff in the catchment of Coomalbidgup Swamp is generated after several months of high rainfall rather than by a single wet month or individual high intensity storms of shorter duration. As the water table rises and saturated areas within the catchment increase, an increased proportion of rainfall is expected to runoff. As a result, there is increasing pressure to implement water management programs on agricultural land within the catchment (Anon. 1990). Diversion of excess water into natural water courses downstream, via a system of contour banks and drains, is one means used to control salinity and waterlogging. Wetlands therefore, are receiving much more water and salt than under natural conditions.

Methods

Species Distribution and Mortality - Transects

Permanent belt transects were established at 4 sites around Coomalbidgup swamp in June 1990 (Fig 1) to represent the different plant communities present. Each transect was 30m x 2m with one end at the water's edge; both ends were marked with a star picket. Changes in elevation along the transects were determined at 2m intervals using a dumpy level. For comparison between swamp and terrestrial vegetation, each transect was extended 50 - 60m into the lake to include inundated trees and shrubs on the swamp bed. Due to the depth of water, the identity of dead understorey species on the deepest part of the transect could not be recorded.

Vegetation physiognomy, species presence and mortality were recorded at 2m intervals along the transects during June 1990 and June 1992. Sampling in June 1990 took place sufficiently close to the time of death to determine the identity of most species. Specimens were collected of all species and identified at the Western Australian Herbarium.

Species Distribution, Vigour and Mortality - Traverse

A permanent transect traversing the swamp was established between transects 1 and 2 in June 1990 (Fig 1). A compass bearing of 145° (from transect 1) was followed by boat and each tree that occurred within 1m either side of the boat was tagged, species recorded, trunk diameter at the height of the tag (1 m above 1990 water level) and tree height measured and vigour determined. Water depth was recorded at approximately 20m intervals or at major changes in tree composition/distribution. Understorey species were not recorded due to the water depth. The traverse was re-visited during June 1992 and the vigour of all tagged trees determined.

Vigour was estimated using a scale of 1 to 5; 1 = dead for >1 year, no leaves; 2 = many dead/dying branches, few green leaves visible; 3 = visible signs of stress but 50 - 70% green canopy; 4 = visibly healthy with few signs of stress, 70 - 90% green canopy; 5 = healthy, no signs of stress, 90 - 100% green canopy. A vigour class of 1.5 was assigned to trees which appeared to have died recently (within the last year), judging by the persistence of dead leaves. Height was determined with an extendable tree measuring pole. Trunk diameter at tag height was measured with a diameter tape.

Seedling Plots

Two seedling plots were established in June 1990 to determine the density of seedling recruitment on the banks of the swamp. Both plots (S1 and S2; Fig 1) were situated close to the swamp shore and orientated perpendicular to the shoreline. The size of the seedling transects varied according to the density and distribution of seedlings; S1 was 15 x 10m and S2 was 10 x 6m. The sites chosen for the transects were representative of recruitment observed in open shrubland (S1) and closed woodland (S2). Recruitment only occurred along the flotsam line(s) on the shore. Each plot was divided into 1m^2 cells within which the density and maximum height of each seedling species was determined.

Results

Transects

The species present and their distribution at each of the transects is shown in Fig 2a - d. Transect 1 vegetation at June 1990 varied from *Banksia speciosa* low (4-5m) woodland on the upper and lower slopes of the present shore, to *Eucalyptus occidentalis* tall (12-14m) woodland on the lake bed (Fig 2a). Understorey growth on the shore was dense, the dominant species being *Jacksonia spinosa*. Apart from a few deaths of *J. spinosa*, more likely due to insect attack or 'natural' senescence, there were no apparent deaths due to flooding on the upper slope of the shore. The lower part of the shore and the nearshore shallows displayed a high mortality of species present before flooding, though *Amphipogon* sp. and *Patersonia occidentalis* were relatively tolerant, and the introduced species *Cirsium vulgare* and *Solanum nigrum* invaded the saturated soils. The presence of *E. occidentalis* seedlings near the June 1990 shoreline, suggests a transition from scattered wetland vegetation on the swamp bed to a littoral community. The presence of dead *B. speciosa* in the nearshore shallow water indicated that the dryland sandplain vegetation once extended below the June 1990 shoreline. On the lake bed itself *E. occidentalis* was the only species observed.

The June 1992 water level was approximately 2 metres lower than in June 1990. Due to the death of the original trees and shrubs and the moist sediment, the area submerged during 1990 but exposed in 1992 represented ideal ground for weed invasion and recruitment of wetland species. As a result, a significant increase in the density of weed and herbaceous native species was recorded on the exposed sediment in June 1992. Species such as *C. vulgare, Hypochaeris glabra, Sonchus oleraceus* and *Eragrostis curvula* were prevalent on the damp exposed shoreline. There was also extensive recruitment of *E. occidentalis* and *Melaleuca cuticularis* seedlings between 1990 and 1992 as the water level receded.

The shore vegetation of Transect 2 (Fig 2b) was typified by a mid-high (0.5-1m) shrubland with emergent *E. occidentalis* and *M. cuticularis* trees. Although there was no structurally dominant species in the understorey, common species were *Grevillea nudiflora, Leucopogon spp.* and *Hibbertia hypericoides.* The slope of the shore at Transect 2 was shallower than that of Transect 1 indicating wetter conditions over a greater area of the transect. This is supported by the presence of *M. cuticularis* on the mid to lower slopes of the shore. Vegetation on the lake bed consisted of *M. cuticularis* mid-high woodland, with dead *Beaufortia* sp., *Acacia alata* and *Hakea laurina* closer to the shore. Shrubs common on the upper slopes of the shore (*e.g. Micromyrtus elobata* and *G. nudiflora*) showed a high degree of mortality when present on the lower flooded areas. Live plants of the lower shore and near shore areas were typically those tolerant of waterlogged conditions, such as *E. occidentalis* and *M. cuticularis*; the introduced species *S. nigrum* also invaded the saturated sediment. The pattern of tree deaths on the lake bed was similar to those of Transect 1. From these remnants of dead vegetation in the nearshore region, it is evident that the dryland vegetation extended 20-30m further downslope of the June 1990 water level. Transect 2 in June 1992, like Transect 1, displayed extensive weed (*C. vulgare, S. oleraceus, Rumex acetosella*) establishment in the area that was submerged in 1990. Both *E. occidentalis* and *M. cuticularis* seedlings had also established over most of this area.

The vegetation of Transect 3 (Fig 2c) was similar in structure and dominant species composition to Transect 1. A *B. speciosa* low (3-6m) open woodland covered the shore and a *E. occidentalis* tall woodland covered the lake bed. Dominant shore species were *Leptospermum erubescens* and *Melaleuca thymoides.* There was high mortality amongst all the species in waterlogged and flooded parts of the transect except seedlings of *E. occidentalis* and *M. cuticularis*, and the introduced species *C. vulgare* and *S. nigrum* which probably established soon after water levels began to recede. The sandplain vegetation had extended 20-30m further downslope of the June 1990 water level. The mortality amongst the *E. occidentalis* on the lake bed was similar to the other transects. Transect 3 in June 1992, like all other transects, had extensive weed (*C. vulgare, S. oleraceus, E. curvula*) and native herb (*Senecio quadridentatus, Epilobium billardierianum, Dampiera sericantha, Muehlenbeckia adpressa, Juncus pallidus, Velleia trinervis*) establishment in the area that was submerged in 1990. Both *E. occidentalis* and *M. cuticularis* seedlings had also established over this area.

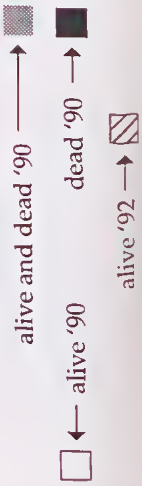
Vegetation on Transect 4 (Fig 2d) was similar in structure to Transect 2, with the understorey being mid-high shrubland covered by scattered emergent *E. occidentalis* and *M. cuticularis.* However, the dominant shrubs of the understorey (*Hakea corymbosa, Phymatocarpus maxwellii* and *A. sulcata*) were different to those of Transect 2. Emergent *E. tetragona* and the shrubs *Acacia subcaerulea, Hibbertia acerosa* and *Melaleuca striata* were only found at the uppermost, and driest, part of the transect. The pattern of shrub and tree deaths was also similar to Transect 2. Transect 4 in June 1992 had weed (*C. vulgare*) and native herb (*S. quadridentatus, E. billardierianum, D. sericantha, V. trinervis, E. monostachya*) establishment in the area that was submerged in 1990. Both *E. occidentalis* and *M. cuticularis* seedlings also established over most of this area.

At all the transects, small seedlings of *E. occidentalis* and/or *M. cuticularis* were found within 2-6m of the shore in June 1990 and 1-26 m of the shore in June 1992. These seedlings were of varying age and had germinated between winter 1989 and autumn 1992. Generally the age of the seedlings increased with increasing elevation.

Traverse

The lake is a flat-bottomed basin with gently sloping sides and had a maximum depth of approximately 5m during the 1990 sampling period.

Figure 2. Profile diagrams and species distribution along shore transects 1-4. Height and distance are to scale.



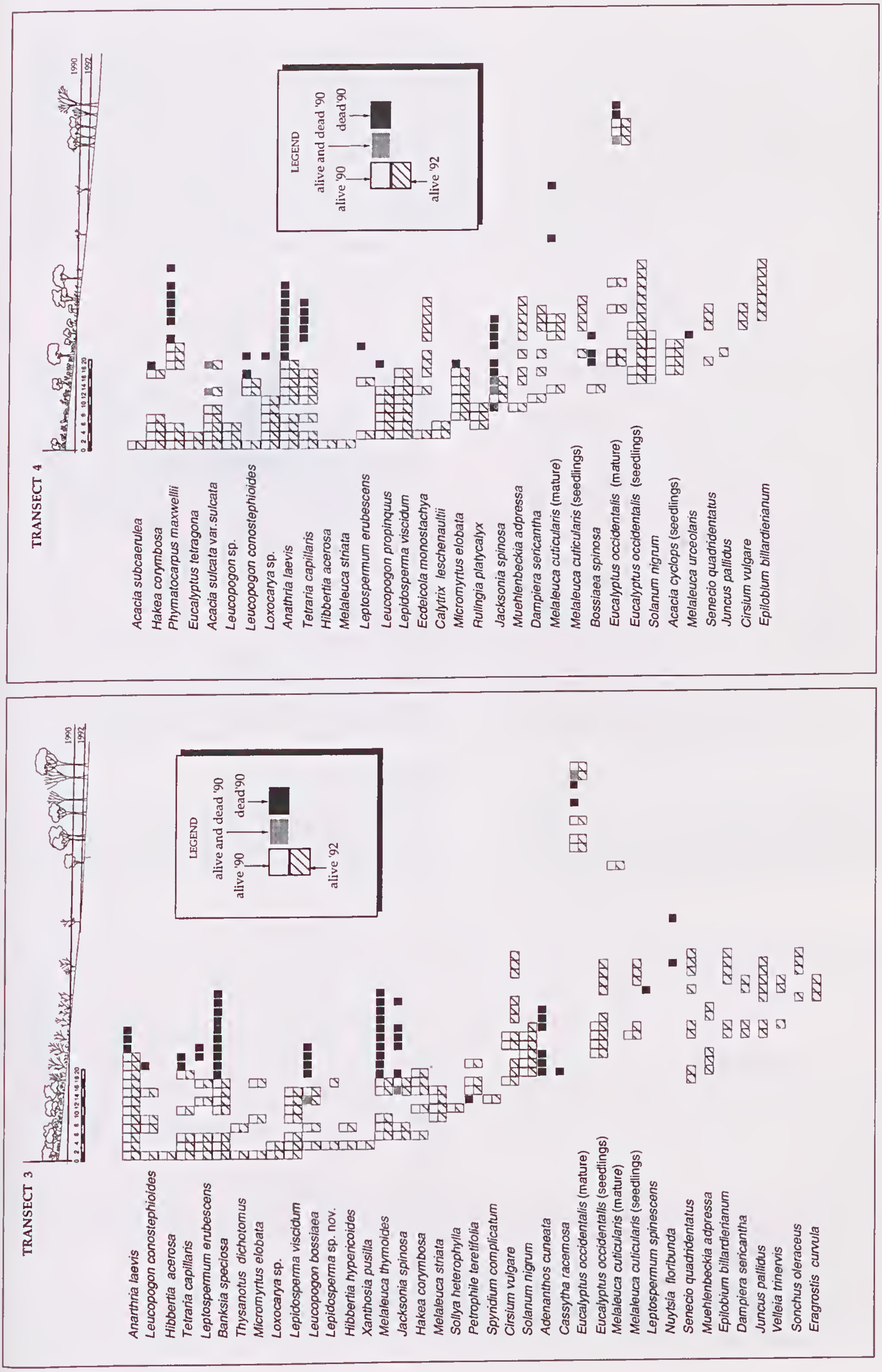


Figure 2.—*continued*

Table 1 shows the vigour classes of the trees sampled and the mean trunk diameter of each class. A total of 173 *E. occidentalis* and 18 *M. cuticularis* plants were recorded along the 1100m transect. It was noted that tree deaths (*E. occidentalis*) along the transect could be classified into two groups, those that had recently died (≤ 1 year ago), and those which had been dead for a much longer period of time. The latter group consisted of 107 (61.8%) trees which perished in a fire in December 1976 (neighbouring landowners, *pers. comm.*). This was evident in the lack of bark and minor branches, weathered trunk and absence of dead attached leaves. Even though local residents report that the lake has not dried since the 1986 flood event, post-1976 tree deaths were very recent (≤ 1 year before sampling). This implies that the inundation between 1986 and 1989 was not sufficient to cause tree death within that period, and that the recent mortality observed on the lake bed is a response to the cumulative effect of the significantly higher water levels after the 1989 flood event. Mortality of the dryland vegetation is also likely to be due to the 1989 flood, as there is no evidence to suggest this vegetation was inundated after the 1986 flood. All recent deaths and poor vigour of the lake bed trees were assumed to be due to prolonged inundation since 1986. Of the remaining live trees most were found to be in poor to fair health (vigour class 3 & 4). In June 1990, trees that had died or were dying since flooding (vigour classes 1.5 and 2) comprised 21.2% of trees alive prior to flooding (Table 1). The June 1992 monitoring of the trees showed that the proportion of dead and dying trees increased to 45.5% of those trees alive prior to flooding. Most live trees remained in the poor to fair health categories, however, there was an increase in the number of dying trees (from 5 to 9) and poor health trees (15 to 24). There was a decrease in the number of fair health trees (from 31 to 12) and healthy trees (from 6 to 0), and the number of live trees (classes 2, 3, 4 and 5) decreased from 55 (83.3%) to 45 (68.2%).

Apart from an area of small diameter trees in the last 250m of the traverse (SE end), there was no apparent pattern of distribution of vigour and diameter classes, and height across the transect. Figure 3 shows the trunk diameter classes versus vigour categories for the trees on the traverse. Most trees sampled were of small to medium diameter, 0 - 16 cm. However, the numbers in these classes were made up significantly by trees that perished in the 1976 fire (Dead category = vigour class 1; Fig 3). This indicates that at the time of the fire a large proportion of the population died, and the survivors have matured since, increasing the range in diameter classes. Trees that had recently died or were dying as a result of prolonged flooding (Recently Dead category = vigour class 1.5 + 2), were mostly medium to large in diameter (up to 26-28 cm) but with a significant number of small diameter trees. Live category trees (vigour class 3 + 4 + 5) had the widest range of diameter classes from 2 cm to 36 cm, with most less than 22 cm.

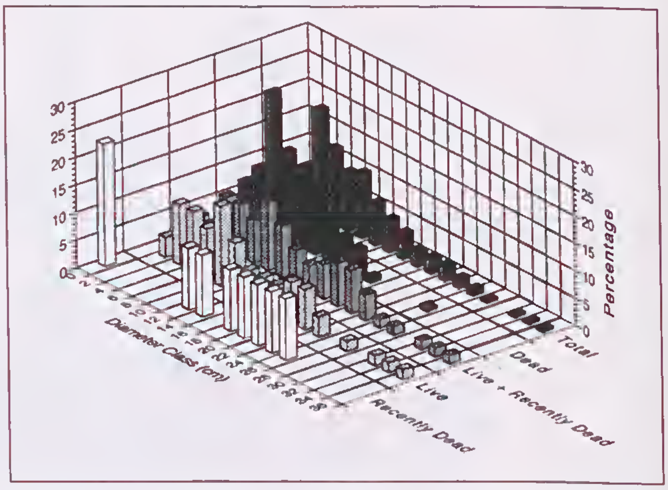


Figure 3. Proportion of trees in each trunk diameter class for different vigour categories. Vigour category Dead = vigour class 1; Live and Recently Dead = vigour class 1.5+2+3+4+5; Live = vigour class 2+3+4+5; Recently dead = 1.5+2. See methods for definition of vigour class.

Seedling Plots

Seedlings were found amongst the flotsam (dead leaves, capsules, seeds) that collected in a series of lines along the shore of the swamp. It is suggested that the trees on the lake bed may have flowered and dropped seed on the water

Table 1

Number (N) and percentage of total trees in each vigour class, and trunk diameter (mean, standard error and range) of trees sampled along the traverse (see Fig 1) for 1990 and 1992

VIGOUR CLASS	1990		1992		1990 TRUNK DIAMETER (cm)		
	N	% total	N	% total	MEAN	S.E.	RANGE
ALL CLASSES	173		173		10.43	0.54	1-36
1 (Dead)	107*	61.8	125	72.3	7.93	0.45	1-26.8
1.5 (Recently Dead)	9	5.2	3	1.7	16.22	3.25	1-26
2 (Dying-very Poor Health)	5	2.9	9	5.2	6.68	1.50	3.1-13
3 (Poor Health)	15	8.7	24	13.9	13.19	1.90	5.4-28.5
4 (Fair Health)	31	17.9	12	6.9	16.55	1.37	5.9-34
5 (Healthy)	6	3.5	0	0	14.92	5.87	4.8-36
Number of trees alive prior to flooding	66						
Number of Recently Dead+Dying Trees	14		30				
(% of trees alive prior to flooding)	(21.2)		(45.5)				

*trees died during 1976 fire prior to flooding surface each year since the 1989 flood. Measurements of seedling density in June 1990 show variation in seedling numbers in relation to the distance from the water's edge, with peak density at the flotsam lines (Fig 4). At the time of sampling, seedling height on the upper flotsam line (Plot 1) was up to 30 cm, and less than 5 cm on the lower flotsam line would have germinated in Autumn 1990 (just prior to the June 1990 sample date). Only a small proportion of the seedlings observed were *M. cuticularis* (Plot 1 only). This is probably due to the relative scarcity of this species at the swamp combined with the timing of seed release. The higher seedling density in Plot 1 may be due to the very open tree and shrub canopy and presence of larger areas of bare ground. At Plot 2, where seedling density was lower, the tree and shrub canopy was dense, therefore restricting light, and the ground was covered with a deep litter layer. Although the seedling plots were not monitored in 1992, it was noted that further flotsam lines developed at lower elevations as the water level receded. This resulted in up to 4 lines (1989-92) of seedling recruitment decreasing in age from high elevations to lower elevations. However these lines were not pronounced where dense vegetation/debris broke up the flotsam. Although not measured, a significant increase in *M. cuticularis* seedlings on the swamp shore was noted in June 1992.

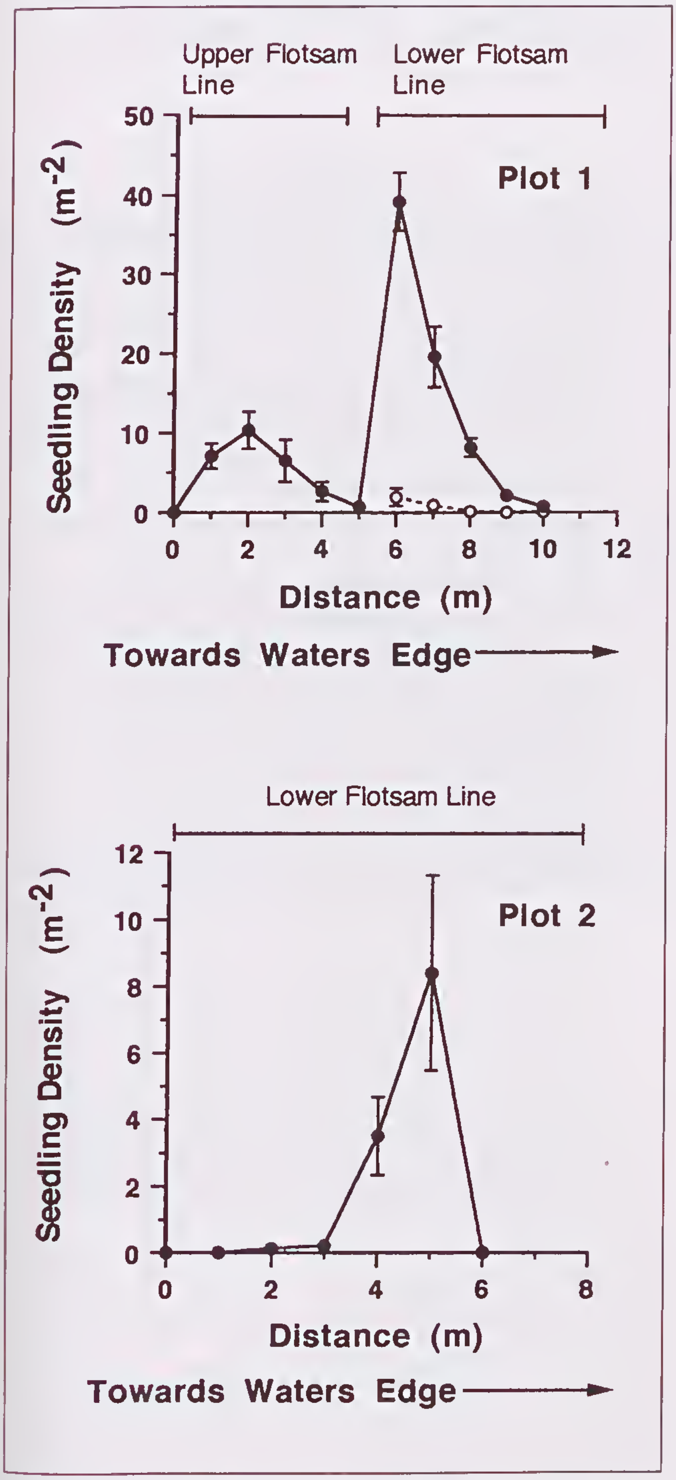


Figure 4. Seedling density along seedling plots 1 and 2. Values are mean ± standard error. ● = *Eucalyptus occidentalis*, O = *Melaleuca cuticularis*

Discussion

The results of this study document the detrimental effect of abnormally high lake levels and prolonged flooding. In all four transects, a high mortality was observed for all species typical of the surrounding dryland sandplain flora. The duration and depth of flooding that resulted in plant mortality is difficult to determine. Local farmers recall that the swamp "filled" during heavy rains of 1986 and has not dried, or had substatial lowering of water levels, since. If this is the case, then plants at lower elevations on the shore transects would have been partially inundated or submerged for up to 4 years by 1990 and 6 years by 1992. However, the evidence suggests that plant mortality at higher elevations on the shore transects occurred ≤ 1 year before June 1990. Those plants at lower elevations on the shore transects (*i.e.* totally submerged in 1990) may have died relatively soon after inundation in late 1986 or 1987. The water depths where dead plants were found in June 1990 varied from waterlogged (up to 50 cm upslope of the June 1990 level) to 3m depth.

As a consequence of the deaths of dryland species, the width of the remaining peripheral dryland vegetation at some parts of the swamp was reduced by one half. Such vegetation is vital to the conservation of wetlands because it acts as a buffer to disturbance and runoff, and is often the only intact native vegetation and habitat for fauna in agricultural areas.

More than 45% of the trees on the lake bed that were alive at the time of the 1989 flooding were either dead or dying by 1992 as a result of prolonged flooding. Mortality of the lake bed trees is expected to increase if the lake does not dry out regularly (every 1-2 years).

On all transects, *M. cuticularis* and *E. occidentalis* were the only perennial species that displayed flooding tolerance. Both species occur naturally in areas subject to inundation, but the duration of flooding under normal conditions would be much shorter (a few months of the year) than experienced under the elevated water regime of Coomalbidgup Swamp.

Depending on future changes in water regime, all or part of the peripheral sandplain vegetation is expected to regenerate gradually. However, from the differences observed between the two sample dates, species composition and diversity of the regenerated vegetation are expected to differ from the pre-flooding condition. Species such as *B. speciosa* require a fire to trigger seed release, and unless this occurs, recruitment is unlikely. Open disturbed areas, caused by death of vegetation due to flooding, were rapidly colonised by weed species, and dense weed growth may severely restrict recruitment and establishment of native species (Hobbs 1988; Hobbs & Atkins 1988). In some areas where seedlings of *E. occidentalis* germinated amongst dead sandplain vegetation, future survival of the seedlings may depend on an elevated water regime. As noted by the June 1992 observations, seedlings of *E. occidentalis* and *M. cuticularis* continued to establish as the water level receded. With continued inundation and subsequent mortality of the trees on the swamp bed, the distribution of these species may become limited to the littoral zone.

At present, Coomalbidgup Swamp contains relatively fresh to brackish water (≤ 4 ppt). Given that water levels are relatively high, this suggests that with lower water levels, salinity will be greater than 4 ppt. Upon drying, salt in the surface soil of the swamp bed may reach a concentration which will adversely affect the surviving vegetation. Furthermore, with continued groundwater rises and increased runoff from the surrounding catchment, the potential for secondary salinisation is significant. Subsoil salt store levels in the Coobidge Creek area are thought to be high (Bob Nulsen, *pers. comm.*), adding to the future threat of salinisation. Although *E. occidentalis* is relatively tolerant of salinity and waterlogging (Van der Moezel *et al.* 1991), higher salinities are expected to have a detrimental effect on the vegetation of the lake bed and periphery. The adverse effects of higher water levels (without periods of drying) and potential increases in salinity on wetland vegetation in south-western Australia has been documented (Froend *et al.* 1987; Bell & Froend 1990; Froend & McComb 1991). If the disturbance in the water and salt balance of the catchment goes unchecked, the result is a gradual degradation of the low-lying wetland areas. It is clear that at Coomalbidgup Swamp, the death of a significant proportion of the surrounding mature sandplain vegetation occurred during 1986-1992 because of abnormally high water levels and prolonged flooding. Judging from the age of the trees that died during the recent flooding, similar water levels (and duration of flooding) have not occurred within 15-20 years before 1986. Under pristine catchment conditions, flooding events of this magnitude would be rare (Anon 1990).

As studies elsewhere have indicated (Froend & McComb 1991), regular flushing (outflow) of Coomalbidgup Swamp during winter and spring would decrease its total salt load. To reduce mean water depth, ensure regular drying, and increase through-flow, the drainage of the lake may be improved by lowering the elevation of the outflow channel. However, the detrimental effects of increased discharge on other wetlands downstream would need to be determined. Other remedial measures, aimed at the cause rather than the symptom, may include better water management practices on agricultural land such as improved retention of water on farms coupled with greater water use through transpiration. Management plans for wetlands of conservation value in the Coomalbidgup area should consider the effects of increased surface runoff, elevated groundwater, and associated transport of nutrients and salt, and means of reducing them.

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